Rapid Assessment of Intertidal Wetland Sediments

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Rapid assessment of intertidal wetland sediments: I. Approach

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Abstract

Urbanization of coastal areas poses a severe threat to ecologically valuable intertidal wetlands. This paper presents a pragmatic approach called Rapid Assessment for Intertidal Wetland Sediments (RAITWS) for evaluating the sediment quality of intertidal wetlands. RAITWS involves construction of reference groups, selection of a subset of environmental variables, matching of test sites to reference groups, prediction of the benthic fauna community structure (e.g. of macroinvertebrates) at test sites, evaluation of the Observation to Expectation ratio (O/E ratio), quantification of environmental variables with series of dynamic numerical models and interpretation of the O/E findings. The proposed method extends the existing rapid biological assessment approach from static to dynamic applications. In particular, RAITWS provides a fast method of assessing intertidal wetland sites which are undergoing ecological change due to nearby coastal development.

Keywords: Rapid assessment; Intertidal wetland; Sediments; Dynamic models; Environment

Introduction

The intertidal zone provides abundant benthic community habitats which help support coastal ecosystems. In recent years, the rapid urbanization of coastal regions has severely impacted on the sustainability of intertidal wetlands (Roman et al., 2000; Thompson et al., 2002). Proper evaluation of sediment quality is necessary in order to monitor the ecological health of intertidal sites. Conventional assessments of sediment quality in wetland systems have focused primarily on physical and chemical variables (Del Valls et al., 1998). But biological components in sediment are more sensitive and responsive to changes in the environmental conditions (Diaz et al., 1993). Assessment based solely on physical and chemical data is therefore not comprehensive, nor sufficiently precise in identifying the interrelationship between sediment and its environment. Instead, biological assessment provides a direct measure of the state and functionality of an aquatic community of plants and animals, and can be used to benchmark environmental quality management programs. Consequently, biological assessment of environmental quality is routinely applied nowadays (Borja et al., 2000; Llanso et al., 2009; Archaimbault et al., 2010). Recently, Comte et al. (2010) devised an efficient approach using an artificial neural network for determining the relationship between biological communities and environmental factors. Comte et al. applied the ANN method to a case involving cause-effect relationships between ecological state and environmental pollution at basin scale, with aquatic invertebrates used as the biological indicator.

From the 1980s onwards, Rapid Biological Assessment (RBA) became widely used to evaluate river biological quality for water quality management (Wright et al., 1984; Chessman, 1995; Barbour et al., 1999). RBA assumes that similar features and structures of biological communities occur at different undisturbed locations characterized by the same environmental parameters and having relatively similar types of habitat (Wright, 1995). Common RBA indicators include periphyton, macroinvertebrates, and fish (Harrison and Whitfield, 2004; Bonada et al., 2006; Moulton et al., 2009). In practice, the RBA method has been found to be efficient, effective, low cost, and easy to apply. Examples of the application of RBA to the evaluation of assemblages of macroinvertebrates are given by Resh & Jackson (1993), Metzeling et al. (2003) and Buss & Vitorino (2010).

In practice, the environmental variables for RBA are usually measured using field surveys. However, comprehensive field measurements are very difficult to obtain for intertidal wetlands which are affected by tidal dynamics. Fluctuations in temperature, salinity, dissolved oxygen, and water content are more extreme in the intertidal region than in the sublittoral zone (Hayward, 1994). Furthermore, the environmental variables in intertidal wetlands are greatly affected by pollution and sudden changes to the environment (Mazik and Elliott, 2000; de la Huz et al., 2005; van der Wal et al., 2008b). Field monitoring is simply too expensive in practice for the dynamically changing conditions of intertidal wetlands. Herein, it is proposed that series of classical models, including hydrodynamics, sediment transport, bed deformation, and water quality models, are used to simulate the dynamic processes in the intertidal zone.

Although a wetland system (such as a marsh or a swamp) is different environmentally from an ordinary water body system (such as a lake or river), the principles inherent in the Rapid Biological Assessment (RBA) method for water bodies can be similarly applied to the assessment of sediment quality. The present paper proposes an approach for Rapid Assessment of Intertidal Wetland Sediment (RAITWS) based on the principles of RBA. In comparison with previous methods, the main advantage of the proposed RAITWS approach is its speed when assessing intertidal wetland sediments. Furthermore, the proposed RAITWS approach is capable of reflecting possible changes in environmental variables by incorporating a series of dynamic models, and so is particularly useful for assessment of intertidal wetland environmental quality in cases where ecological changes are taking place due to nearby coastal development.

The paper is structured as follows. The Methods Section outlines the RAITWS methodology. The Discussion Section describes the principles behind the methodology, and its implementation. Conclusions and recommendations are listed in the final Section.

Methods

The RAITWS methodology is based on the principles of the RBA method. The core steps of RAITWS (Fig. 2) involve (i) selection of reference sites based on a non-disturbance (or minimal level of damage) principle, (ii) classification of reference groups for different sites according to the uniformity of bio-community structures, (iii) assessment of sediment quality variation by comparing the observed bio-community characteristics at test sites with reference data (i.e. the Observation to Expectation O/E ratio), and (iv) use of numerical model simulations to provide data by which to interpret the (O/E) ratio. Herein, the bio-communities comprise benthic macroinvertebrates.

Construction of Reference Group

The key to success in using RAITWS is the careful selection of reference sites. Such sites should comprise wetlands (e.g. salt marsh) which have been largely unaffected by human activities and have little to no invasive land use within their 1000m zone of influence (Carlisle et al., 1998). High natural variability may be encountered while characterizing the reference sites, making it difficult to distinguish between anthropogenic and environmental effects on biotic communities. Therefore, it is essential to minimize the natural variability within the monitoring network in order to allow the impacts of anthropogenic activities to dominate. In practice, there are two ways to minimize the natural variability: One common approach is to organize environmental information on a narrow spatial scale; another is to use typology characteristics to group similar patterns of wetland. Reference sites could be established by several means: the collection of in situ data, the use of historical data, employing predictive models, or from expert judgment (Barbour et al., 1994). Consider *in situ* data for example. Based on observed biological data, the reference sites can be classified into groups according to their uniformity by means of multivariate statistics. Currently, the aquatic organisms considered in RBA mainly comprise benthic invertebrates, algae and fish. In wetland assessment, the most frequently used biological indicators are benthic macroinvertebrates because their relatively long life cycle, more stationary behavior, and occupation of a wider niche in the food web constitute a fairly complete biological accumulation process.

Selection of a Subset of Environmental Variables

Discriminant function and correlation analyses are used to select a subset of environmental variables relevant to the determination of the reference groups. Primary environmental variables of RAITWS for wetland sediment are descriptive indicators closely related to the biological community structure of the wetland, such as latitude, longitude, pH, salinity, water saturation, inundation time, tidal range, total organic carbon, and sediment size grading. Variations between the reference groups are explained by a subset of these indicators. In practice, it makes sense to select 5 to 15 variables for each subset (Moss et al., 1987).

Matching Test Sites to Reference Groups

Multiple discriminant analysis (MDA) is carried out on the environmental variable data, and the results used to determine the reference groups into which the test sites fall, after which the probabilities are calculated of the test site falling into each reference group.

Estimation of Benthic Fauna Community Structure at Test Sites

The benthic fauna community structure at each test site is estimated using the following procedure. (1) Calculate the probabilities of any given (new) sampling site falling into each reference group. (2) Calculate the occurrence probability of prescribed species to exist at a given site. This involves multiplying the percentage frequency of the prescribed species present in each reference group by the probability that the given (new) sampling site falls within the corresponding reference group, and then summing these products. (3) Estimate the integrated occurrence probability (an important prediction index) by summing the occurrence probabilities of the prescribed species over the entire set of reference groups. (4) Restrict predicted species under consideration to those of > 50% integrated occurrence probability (Moss et al., 1987).

To quantify the amount of predominant species, the number of individuals related to each species is first determined. The degree of similarity between different sites is determined by the presence of common species (Norris and Norris, 1995; Resh et al., 1995). Rare species obscure data integrity causing difficulty in interpreting the results of multivariate analysis. In practice, it is recommended that any species with an occurrence rate less than 10% should be excluded.

Computation of O/E (Observation/Expectation) Ratio

An Observation to Expectation (O/E) ratio in the range from 0 to 1 is used to evaluate the state of the benthos community structure, with 1 indicating that conditions in the observed community correspond to exactly those expected. Expectation equals the sum of occurrence probabilities of all the predicted species of organisms; whereas Observation denotes the number of species that are found.

Interpretation of Sediment Quality Changes

The deviation of O/E (Observation/Expectation) ratio indicates whether the study area has been disturbed by human beings. However, it is difficult to explain the variation in O/E ratio between different sites. Given that it is well established that the structure of a benthos community is closely tuned to the local environmental conditions (Clarke, 1993; Sponseller et al., 2001), analysis of changes to individual environmental variables (e.g. depth of habitat, distance from shore, latitude and longitude, marine and hydrological conditions, physical and chemical properties of bed sediments, and heavy metal contamination) related to benthic fauna provides an effective means of explaining the variation of community structure in terms of environmental factors. Intertidal wetlands have highly sensitive environments, which are affected by tidal dynamics that change cyclically the physicochemical characteristics, including salinity, water temperature, and contaminant concentration. Conventional techniques for environmental monitoring are difficult (and very expensive) to apply to the measurement of the dynamic processes affecting intertidal wetlands. Instead, we take a similar approach to that commonly adopted in water resources, and use series of dynamic numerical models to simulate the dynamic processes in an intertidal wetland, and hence characterize quantitatively certain environmental variables. The results from the dynamic models are helpful in providing a reasonable interpretation of the variation in O/E ratio between different sites.

The dynamic models should include hydrodynamics, sediment transport, bed deformation, and water quality modules. In practice, the intertidal wetland can be assumed to be a shallow system, and so the model could be two-dimensional horizontal (2DH). Without loss of generality, a typical set of depth-averaged governing equations for the water-sediment-contaminant system are given below. The mass and momentum of the flow hydrodynamics may be expressed as (Leendertse, 1987)

$$\frac{\partial \zeta}{\partial t}+\frac{\partial (Hu)}{\partial x}+\frac{\partial (Hv)}{\partial y}=0, \tag{1}$$

$$\frac{\partial u}{\partial t}+u\frac{\partial u}{\partial x}+v\frac{\partial u}{\partial y}=fv-g\frac{\partial \zeta}{\partial x}-\frac{gu\sqrt{u^2+v^2}}{HC^2}+\frac{C_w\rho_a w^2\cos\beta}{H\rho}+\frac{\partial}{\partial x}(A_x\frac{\partial u}{\partial x})+\frac{\partial}{\partial y}(A_y\frac{\partial u}{\partial y}), \tag{2}$$

and

$$\frac{\partial v}{\partial t}+u\frac{\partial v}{\partial x}+v\frac{\partial v}{\partial y}=-fu-g\frac{\partial \zeta}{\partial y}-\frac{gv\sqrt{u^2+v^2}}{HC^2}+\frac{C_w\rho_a w^2\sin\beta}{H\rho}+\frac{\partial}{\partial x}(A_x\frac{\partial v}{\partial x})+\frac{\partial}{\partial y}(A_y\frac{\partial v}{\partial y}), \tag{3}$$

where $\boldsymbol{\zeta}$ is the tidal elevation above mean water level, $\boldsymbol{H}$ is the instantaneous total water depth, $\boldsymbol{u}$ and $\boldsymbol{v}$ are depth-averaged velocity components in the $\boldsymbol{x}$ and $\boldsymbol{y}$ directions, $\boldsymbol{f}$ is the Coriolis parameter, $\boldsymbol{g}$ is the gravitational acceleration, $\boldsymbol{C}$ is the Chézy coefficient related to bed roughness, $\boldsymbol{C_w}$ is the wind stress coefficient, $\boldsymbol{\rho_a}$ is the density of air, $\boldsymbol{w}$ is the wind speed, $\boldsymbol{\beta}$ is the angle between wind direction and the positive $\boldsymbol{x}$ direction, and $\boldsymbol{A_x}$ and $\boldsymbol{A_y}$ are the kinematic eddy viscosities in $\boldsymbol{x}$ and $\boldsymbol{y}$ directions.

Assuming that conditions are well-mixed in the vertical direction, suspended sediment transport may be described by the following 2D depth-averaged advection`-dispersion equation

$$\frac{\partial (Hs_i)}{\partial t}+\frac{\partial (Hus_i)}{\partial x}+\frac{\partial (Hvs_i)}{\partial y}=\frac{\partial}{\partial x}(D_xH\frac{\partial s_i}{\partial x})+\frac{\partial}{\partial y}(D_yH\frac{\partial s_i}{\partial y})+Q_i \tag{4}$$

where $\boldsymbol{s_i}$ is the depth-averaged suspended sediment concentration of the $\boldsymbol{i}$-th constituent, $\boldsymbol{D_x}$ and $\boldsymbol{D_y}$ are dispersion coefficients in the in x and y directions, and $\boldsymbol{Q_i}$ is the source/sink term for the ***i***-th constituent.

Bed deformation is governed by the following equation (derived from conservation of mass of the bed material)

$$\partial_t (S^i B)_k = -\delta(k,k_t) J^i_{SB} + \alpha_A \delta(k,k_t) J^i_{PA} - \alpha_A \delta(k,k_t - 1) J^i_{PA} \tag{5}$$

where $\boldsymbol{S^i}$ is the mass sediment concentration of per total volume of bed layer $\boldsymbol{k}$, $\boldsymbol{B}$ is the layer thickness, $\boldsymbol{J_{SB}}^{\boldsymbol{i}}$ is the net sediment mass flux, $\boldsymbol{\alpha_A}$ is an armoring parameter (1 for armoring, 0 otherwise), $\boldsymbol{J_{PA}}^{\boldsymbol{i}}$ is the parent to armoring layer when the top or surface layer of the bed, $\boldsymbol{k_t}$, acts to simulate armoring, and $\boldsymbol{i}$ is the ***i***-th sediment constituent.

The surface erosion rate, $\boldsymbol{\varepsilon}$, is commonly characterized by an empirical equation related to the flow-induced instantaneous bed shear stress, given by

$$\varepsilon = \begin{cases} \mathrm{M}' \left(\dfrac{\tau_b - \tau^c_{\mathrm{E}}}{\tau^c_E} \right), & \tau_b > \tau^c_E \\ 0, & \tau_b \le \tau^c_E \end{cases} \tag{6}$$

where $\boldsymbol{M'}$ is surface erosion rate constant, $\boldsymbol{\tau_b}$ is the flow induced bed shear stress and $\boldsymbol{\tau_E}^{\boldsymbol{c}}$ is the critical shear stress for erosion. The deposition rate, $\boldsymbol{Q_D}$, may also be determined empirically from

$$Q_D = \begin{cases} \omega_s s \left(\dfrac{\tau^c_{\mathrm{D}} - \tau_b}{\tau^c_D} \right), & \tau_b < \tau^c_D \\ 0, & \tau_b \ge \tau^c_D \end{cases} \tag{7}$$

where $\boldsymbol{\omega_s}$ is sediment settling velocity, $\boldsymbol{s}$ is sediment concentration, and $\boldsymbol{\tau_D}^{\boldsymbol{c}}$ is the critical shear stress for deposition.

Assuming that there is instantaneous local equilibrium (and hence a constant partition coefficient) between the dissolved and particulate phases, a suitable 2DH equation of water quality model is

$$\frac{\partial(Hc_i)}{\partial t}+\frac{\partial(Huc_i)}{\partial x}+\frac{\partial(Hvc_i)}{\partial y}=\frac{\partial}{\partial x}(D_x H\frac{\partial c_i}{\partial x})+\frac{\partial}{\partial y}(D_y H\frac{\partial c_i}{\partial y})+S_o-K_i c_i \tag{8}$$

where $\boldsymbol{c_i}$ is the concentration of the $\boldsymbol{i}$-th constituent, $\boldsymbol{S_o}$ is the source term; $\boldsymbol{K_i}$ is the decay coefficient of the $\boldsymbol{i}$-th constituent.

In practice, the above system of equations can be solved numerically on a suitable computational grid using any one of an array of discretization techniques, such as finite element, finite volume, or finite difference methods. The important thing is that the model should be validated, be grid convergent and stable, and preferably be computationally efficient. There are also many standard commercial codes (e.g. ADCIRC, Delft3D, and TELEMAC) and open source academic codes (such as HSCTM2D, EFDC) available. Input data required by the model would typically include bathymetry, meteorological data, inflow discharges and sediment and contaminant fluxes, outflow depths and sediment and contaminant fluxes, and external loadings. After calibration to the conditions of a particular site, model performance should be checked by verification.

The interactions among hydrodynamics, sediment transport, water quality, benthic invertebrates and wetland plants may be described as follows (Fig.1). Plants efficiently dissipate wave energy (Massel et al., 1999) and help attenuate currents (Teo et al., 2009). Plants enhance sediment deposition by physically slowing the flow speed of sediment-laden tidal waters (Callaway et al., 1997). Extreme rates of deposition or scouring can have an adverse effect on plant species. Much of the vegetation can be washed away by sediment scouring, and habitat areas will be reduced. Sediment deposition can be more harmful when the sediment accumulation essentially covers marsh and mangrove areas (Rejmanek et al. 1998). The plants are likely to be killed by sediment accretion since oxygen transportation to the roots will be inhibited. Mangrove plants act as a natural filter to trap the suspended particles, excess nutrients and other pollutants. Mangrove vegetation can also be affected by pollution and nutrient enrichment (Ellison et al., 1996; Alongi, 2002). Effects include reductions in growth, biomass accumulation, productivity and foliar nutritional status. Mangrove plants serve as critically important habitats for benthic invertebrates. The density and composition of invertebrate communities are directly influenced by the amount and species of mangroves plants (Strayer et al., 2007). The abundance and diversity of benthic invertebrate populations are also affected by a wide range of physical parameters (e.g. substrate size, depth, dissolved oxygen concentrations, pH, salinity, sediment C/N ratios, and hydrograph) (Alongi et al., 2005; Teo et al., 2009). Previous studies have shown that limited levels of nutrients enrichment in estuaries may result in increased densities of some benthic organisms (Pearson et al., 1978). Moreover, heavy metals and other toxicants derived from agricultural, industrial and domestic sources have detrimental effects on benthic organisms (Saha et al., 2006).

Discussion

The RAITWS approach exploits the ideas behind RBA in order to provide a fast assessment of the interrelationship between sediment and its environment in intertidal wetlands. Compared with the Invertebrate Community Index (ICI) developed by

Hicks (1997) for assessing wetland environmental quality, RAITWS seems more dynamic for intertidal wetland. Its efficiency arises primarily from the assessment processes themselves. Classification of large reference sites into groups and the matching of test sites to reference groups can be rapidly undertaken using standard software (such as the Statistical Package for the Social Sciences: SPSS). RAITWS offers considerable potential as a decision-making tool for use by coastal managers.

In RBA, reference sites are located where the representative bio-communities are under natural conditions or at least do not experience any obvious disturbance. However, 'no-disturbance' sites are rarely encountered, especially when there are local human activities, such as nearby urbanization. For RAITWS the selection criteria for reference sites are broadened by taking the initial state of the sediments in intertidal zone to be the benchmark, and comparing the later state of the sediments against this benchmark to assess changes taking place in sediment quality. This makes RAITWS method appropriate for areas where accelerated economic development has significantly altered the environment.

The environmental variables must be carefully selected. For example, tide, salinity, water temperature, mud thickness, sediment and contaminant concentration (including dissolved and adsorbed phase) are necessary variables to provide the basic ecological template that structures the benthic community (Griffiths, 1991). In this context, many studies have been undertaken towards understanding the relationship between environmental factors and benthic community (Menge et al., 1997; Pusceddu et al., 2007; Schletterer et al., 2010). In both RBA and RAITWS, the environmental variables have two roles: to match test sites to reference groups, and to interpret temporal and spatial variations of the O/E ratio. RAITWS is specifically designed for intertidal wetlands, where the processes are tidally varying, and it is extremely difficult (and costly) to obtain comprehensive field data. Obviously, a field campaign is needed to provide survey information on the bathymetry, and certain measurements for model calibration and verification purposes. The advantage of the present RAITWS approach is that it provides numerical predictions of the temporal and spatial variations of certain key environmental variables using series of dynamic models containing the hydrodynamic, sediment, morphological, and pollutant contamination processes. The numerical model can incorporate the effects of major forcing conditions related to coastal flood inundation due to storm surge or typhoon, sea level rise, and nearshore circulation patterns. Computation of individual environmental variables, and comparison of the results obtained for different sites greatly helps in interpreting the temporal and spatial variation in O/E ratio in terms of the relationship between the benthos macro-invertebrate communities and environmental factors.

Given that benthos community structure appears to have a direct relationship to the environmental state (Warwick et al., 1991; Hedrick et al., 2010), several researchers have tried to use this relationship to forecast the variation of benthos community structure under different environmental impacts. Pinckney and Zingmark (1993) developed and verified a habitat-specific production simulation model to quantify annual benthic micro-algal production in North Inlet estuary, South

Carolina, USA. van der Wal et al. (2008a) constructed response models to predict biomass and species richness of macro-benthos as functions of environmental variables in the Western Scheldt, southwest Netherlands. Yang et al. (2010) developed a three-dimensional hydrodynamic model to simulate estuarine processes in the Stillaguamish River estuary, Washington, USA, and then combined the model results with biophysical measurements to predict habitat responses to a estuary restoration project. In a similar way, RAITWS can also be used to establish the correlation between benthos macro-invertebrate communities with simulated environmental variables, and hence forecast the future trends of intertidal wetland benthos communities under prescribed environmental impacts.

Conclusions

This paper has presented a rapid assessment approach for intertidal wetland sediment (RAITWS), which is an extension of rapid biological assessment for water bodies. The method is designed to be robust, and straightforward to implement in practice. The proposed method extends the static state and absolute value approach of rapid biological assessment method into dynamic applications. Firstly, RAITWS can quickly identify which study sites are exhibiting signs of ecological damage due to human activity. Importantly, a numerical model that couples hydrodynamic, sediment transport, bed deformation, and water quality modules should be used to simulate the tidal dynamic processes affecting the intertidal wetland, and thus provide interpretation of variations in O/E ratio. RAITWS offers an efficient means of assessing sediment quality for intertidal wetlands, and is particularly appropriate for application to coastal areas where accelerated economic development has resulted in changes to the local environmental conditions. The method provides a means of studying intertidal wetlands where there are insufficient data available to utilize conventional sediment quality assessment methods. The companion paper (Lei et al., 2010) demonstrates the application of RAITWS to intertidal wetlands on the coast of Deep Bay, China.

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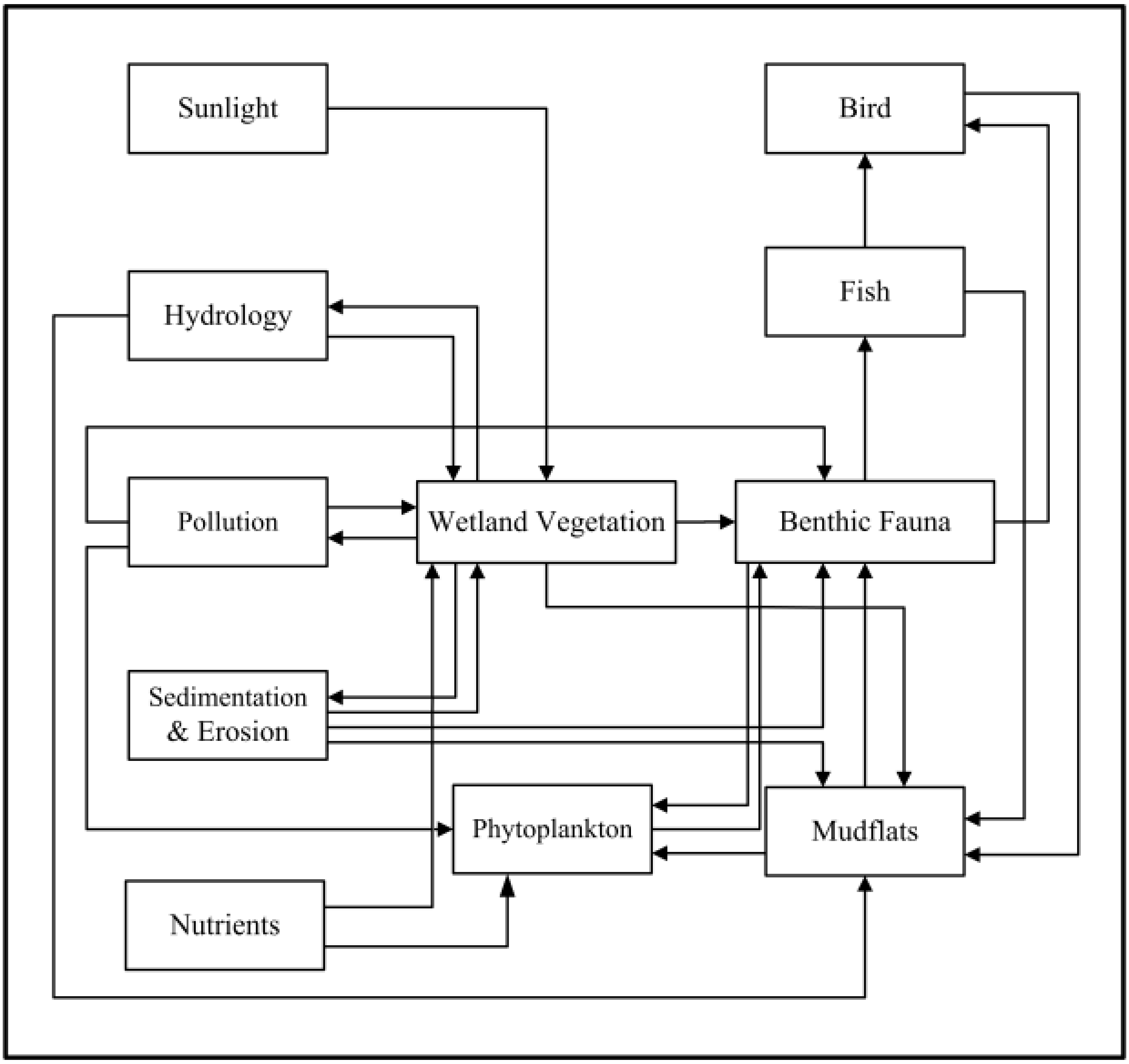
Captions

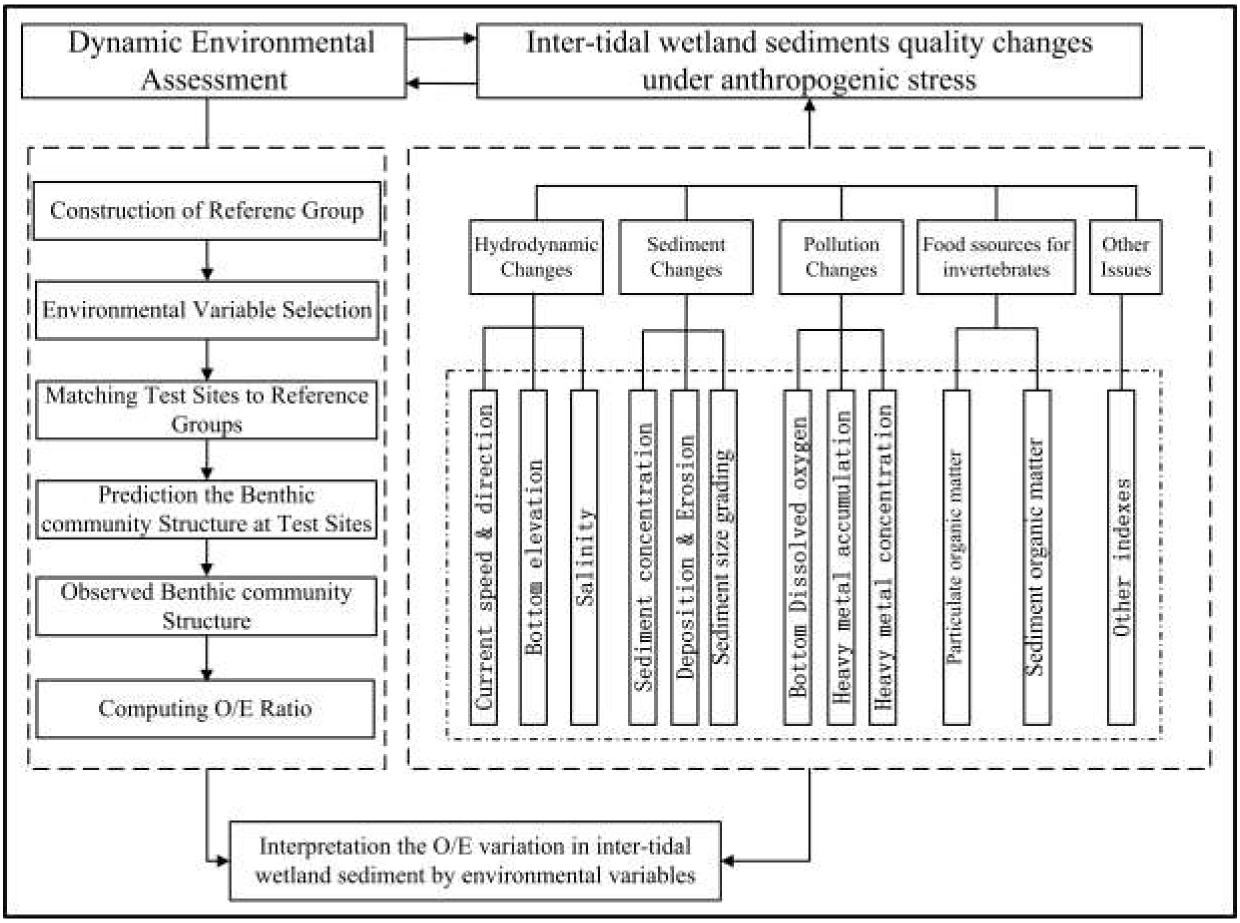
Figure 1. Interactions among hydrodynamics, sediment transport, water quality, benthic invertebrates and wetland plants

Figure 2. Flow diagrams of the RAITWS approach

Table 1 Initial and boundary conditions of the dynamic numerical model

Table 1 lists the initial and boundary conditions for the hydrodynamic, sediment transport, water quality and bed deformation equations. In Table 1, ζ is the tidal elevation above mean water level, $\zeta(t)$ is the monitored tidal elevation above mean water level, $\vec{U}$ is the horizontal flow velocity vector, $\vec{n}$ is the unit normal vector directed out of the closed boundary, s_i is the sediment concentration of the i-th constituent, c_i is the contaminant concentration of the i-th constituent, V_n is the depth-averaged velocity component normal to the boundary during the ebb tide; and B is the layer thickness.


Sunlight
Bird
Hydrology
Fish
Pollution
Wetland Vegetation
Benthic Fauna
Sedimentation & Erosion
Phytoplankton
Mudflats
Nutrients


Dynamic Environmental Assessment
Inter-tidal wetland sediments quality changes under anthropogenic stress
Construction of Referenc Group
Environmental Variable Selection
Matching Test Sites to Reference Groups
Prediction the Benthic community Structure at Test Sites
Observed Benthic community Structure
Computing O/E Ratio
Hydrodynamic Changes
Sediment Changes
Pollution Changes
Food ssources for invertebrates
Other Issues
Current speed & direction
Bottom elevation
Salinity
Sediment concentration
Deposition & Erosion
Sediment size grading
Bottom Dissolved oxygen
Heavy metal accumulation
Heavy metal concentration
Particulate organic matter
Sediment organic matter
Other indexes
Interpretation the O/E variation in inter-tidal wetland sediment by environmental variables